SALT MARSH VEGETATION MONITORING REPORT 2008, CAPE COD NATIONAL SEASHORE

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Executive summary

Cape Cod National Seashore's (CACO) vegetation monitoring program for unrestricted salt marshes was begun in 2003. Prior to that year, salt marsh vegetation monitoring had been focused on tidally-restricted marshes associated with specific tidal restoration projects. In 2008, the vegetation and porewater (salinity and sulfide concentrations) of unrestricted marshes were re-surveyed and the monitoring network was expanded by 150 new plots to improve the spatial resolution of data. In addition, an entirely new marsh site (Jeremy marsh at the southern tip of the Great Island peninsula) was incorporated into the monitoring network.

Comparison of 2003 and 2008 vegetation data and 2003 and 2007 aerial photography showed that broad patterns of vegetation changed relatively little at most marsh sites during this time period. Notable exceptions to this were sites in the Great Island region of CACO (Middle Meadow, the Gut, Jeremy marsh) where salt marsh vegetation losses (also known as salt marsh dieback) have continued at a rapid pace. In these areas there are three main processes occurring: 1) disappearance of cordgrass (*Spartina alterniflora*) throughout the low marsh zone, 2) disappearance of salt marsh hay (*Spartina patens*) along the seaward edge of the high marsh, and 3) landward advancement of *S. alterniflora* into areas previously occupied by *S. patens*. Other changes in marshes unrelated to dieback were due to variability in the abundance of annual forb species and the re-vegetation of barrier-beach overwash areas. With respect to porewater chemistry, salinities and sulfides marshes were very similar in 2003 and 2008. Moreover, the pattern among marshes for each year of sampling was similar.

The completion of this second survey of unrestricted marsh sites allowed for a rigorous analysis of the data and protocol methodology. This exercise made it clear that a number of important factors could contribute the misinterpretation of information if not recognized and handled properly. These factors are discussed in detail in this report and include the following points:

- Temporal changes in vegetation patterns may be overstated at sites with fewer sampling plots.
- Temporal changes in vegetation may reflect subtle changes in the boundaries of plant populations without any corresponding change in overall position in the marsh or area extent.
- Temporal changes in vegetation may be an artifact of plot placement when permanent markers cannot be found.
- Plot monitoring networks may fail to capture large and important changes in marsh landscapes.

Suggestions for various improvements were made to address these issues for future monitoring. One key component of this is aerial image analysis, which provides much needed perspective on trends indicated by plot data.

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Introduction

Salt marsh ecosystems are an important natural resource within Cape Cod National Seashore (CACO), making up roughly 10% of its total area. The benefits of salt marshes are numerous and have been well documented in the scientific literature. In addition to their aesthetic value, they are extremely productive ecosystems that provide critical habitat for a host of plant and animal species (Nixon and Oviatt 1973, Roman et al. 2001). Salt marshes also reduce coastal erosion by dissipating wave energy and ameliorate the effects of nutrient inputs to coastal waters (Bertness 2007).

In 2003, CACO expanded its salt marsh vegetation monitoring program to include the vast majority of unrestricted salt marsh habitat within the Seashore boundary (Smith 2004). Prior to this, monitoring was focused on three tidal restoration projects in Provincetown (Hatches Harbor), Truro (East Harbor), and Wellfleet (Herring River). In 2008, both the unrestricted marsh sites and tidal restoration project sites were revisited and vegetation and porewater data collected. In addition, 150 new plots were added to the monitoring network in order to expand the spatial coverage. The new plots were located at random distances along new, randomly-placed transects. Maps of these new plots can be found in Appendix II.

At present, every unrestricted salt marsh system within the boundary of CACO has permanent ground-level plots for long-term monitoring plots. The following report summarizes the data only from unrestricted marsh sites. Data from the tidal restoration project areas are contained in a separate report (Smith et al. 2009). The majority of this report discusses marsh vegetation change (or lack thereof) between the 2003 and 2008 surveys. Also included is an evaluation of the current methodology and recommendations for the future developmen of the program.

Methods

A detailed description of materials and methods can be found in (Smith 2004), also available online at:

http://www.nps.gov/caco/naturescience/upload/Salt%20Marsh%20vegetation%20Monitoring%20report%202003.pdf.

Study areas

The following sites are included in CACO's salt marsh vegetation monitoring program for unrestricted marshes: 1) West End (WE) (Provincetown; est. 2003), 2) Hatches Harbor (HH) (unrestricted side of dike; Provincetown; est. 1997), 3) the Gut (GU) (Wellfleet; est. 2004), 4) Middle Meadow (MM) (Wellfleet; est. 2003), 5) Jeremy marsh (JM) (Wellfleet; est. 2008), 6) Nauset (NI, NM) (Eastham; est. 2003), and 7) Pleasant Bay (PB) (Orleans; est. 2003) (Figure 1). New plots were added to the West

End and Pleasant Bay marshes. Jeremy marsh was a new site with no previously-established plots.

Figure 1. Map of CACO's unrestricted marsh sites for vegetation monitoring.

Vegetation

One methodological difference between the 2003 and 2008 vegetation surveys was that cover by species in 2008 was assessed visually based on a modified Braun-Blanquet scale (Smith et al., in press). For data analysis, % cover that had been estimated by point-counts in 2003 was converted to corresponding cover class values. The advantages of the visual method over point counts in conducting large vegetation monitoring programs are numerous and justification for its use can be found in Appendix I. Macroalgae, wrack, litter, and standing water cover types were not included in the analyses due to the ephemeral and highly variable nature of these components. In plots where *Spartina alterniflora* was present, the heights of the 5 tallest plants were measured to the nearest cm, as was done in 2003 (Smith 2004).

Although the new plots established in 2008 were situated at random distances along transects while the original plots were spaced uniformly, all transects themselves were located in a random manner. As such, it has been argued by Roman et al. (2002) that plots established by the former method can be considered independent replicates. For the analyses described below, all plots were treated as such. Non-metric multidimensional scaling (MDS) was used to illustrate species composition shifts. Analysis of Similarity (ANOSIM), which is based on Bray-Curtis similarity indices generated from cover class values, was run to determine significance level of community-level shifts (Primer™ ver. 6.0). For single variables, one-way Analysis of Variance (ANOVA) was used to test for significant differences among groups.

Results

Comparison of salt marsh plant communities in 2003 vs. 2008

Although changes, sometimes quite dramatic, occurred at a number of individual plots within each marsh between 2003 and 2008, plant communities as a whole were statistically unchanged (Table 1). Only Hatches Harbor exhibited significant deviation, and this was mainly due to an increase in the cover of annual forbs belonging to the genus *Salicornia*. The Nauset mainland site approached statistical significance (with respect to community change) due to increases in the cover of *S. alterniflora* as it re-colonized a barrier-beach overwash area that was mostly bare in 2003 (Figure 2).

Figure 2. *S. alterniflora* invasion of a plot between 2003 and 2008 in an overwash area of the Nauset mainland site.

Table 1. ANOSIM results comparing plant community composition of marshes in 2003 vs. 2008 (asterisk indicates a statistically significant result).

	Global R value	p value
Hatches Harbor	0.059	0.03*
Nauset Island	0.051	0.09
Nauset mainland	0.068	0.05
The Gut	-0.007	0.52
Middle Meadow	-0.006	0.53
Pleasant bay	-0.004	0.48
West End	0.014	0.12

Middle Meadow and the Gut marshes were not statistically different in 2003 vs. 2008, even though there were large shifts in MDS centroid values (means of x & y coordinates for each site) between 2003 and 2008 (Figure 3). Further analysis of the data showed that this was the result of *S. patens* and *S. alterniflora* losses and the encroachment of *S. alterniflora* into bare areas where *S. patens* had disappeared in certain plots (Figure 4, 5, 6). In essence, large changes in just a few plots were enough to displace the centroid values in ordinal space, but not enough to influence the statistical significance of change across the entire site. When these outlier plots were removed from the analysis, the ordination is transformed quite dramatically, with very little spatial offset between 2003 and 2008 (Figure 3).

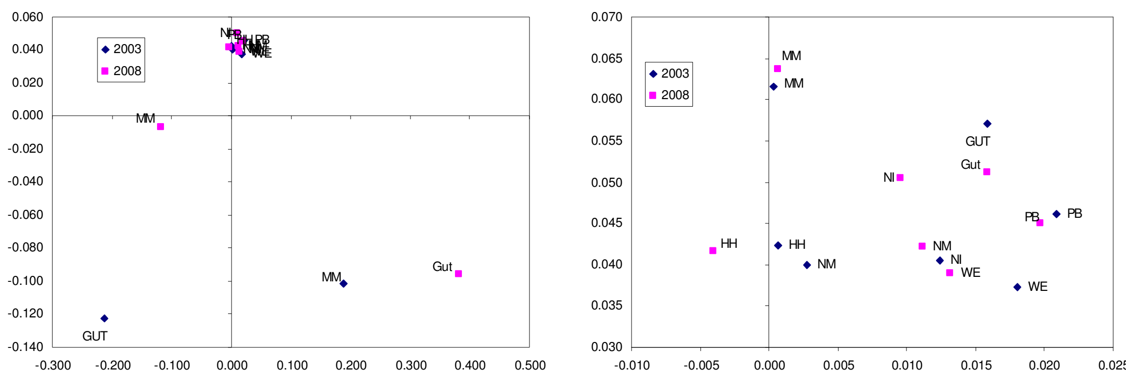
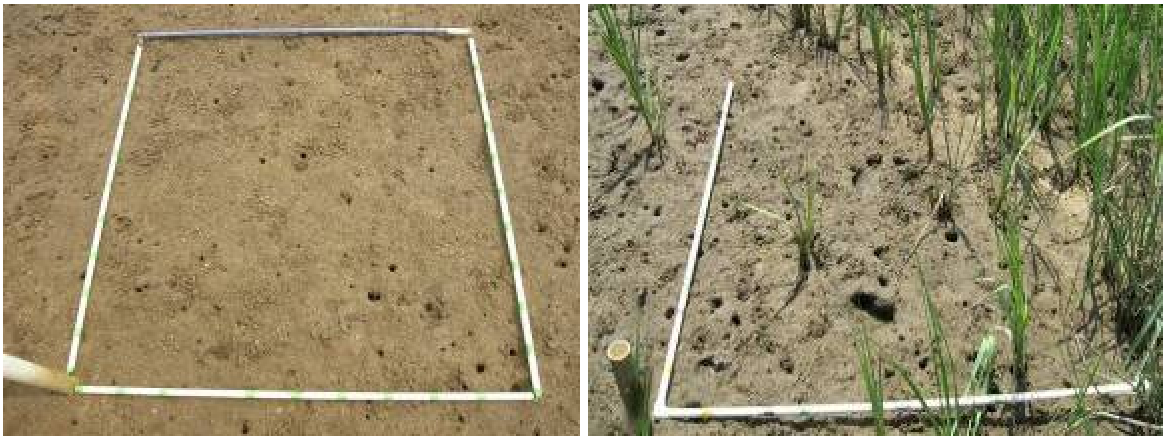


Figure 3. Non-metric multidimensional scaling of plant cover values by marsh and year. Centroids of all plots are shown in the plot on the left while centroids not including plots GU5-100, MM2-60, MM3-000 and MM3-180 are shown on the right.

Note: The loss of *S. alterniflora* at Middle Meadow and the Gut is the result of intense herbivory by a native crab (*Sesarma reticulatum*; purple marsh crab) (Holdredge et al, in press). This can result in major structural changes in marsh architecture as sediments are subsequently lost by erosion (Smith, in press). The disappearance of *S. patens* (and to a lesser extent *D. spicata*) is not yet fully understood, but may be the result of a combination of different factors such as herbivory and other forms of disturbance (e.g., wrack smothering), sea level rise, and soil properties (studies ongoing). (For more information about salt marsh dieback on Cape Cod, please refer to Holdredge et al, in press, Smith, in press; and: http://www.nps.gov/caco/naturescience/salt-marsh-dieback.htm).



2003 MM3-180 2008

Figure 4. Advance of *S. alterniflora* into old *S. patens* dieback area in Middle Meadow.

2003 MM3-000 2008

Figure 5. Loss of *S. patens* in Middle Meadow.

2003 GU3-080 2008

Figure 6. Loss of *S. alterniflora* in the Gut.

In terms of species frequencies (number of occurrences of a particular species/number of monitoring plots at the site), several notable shifts occurred. The high marsh perennial *S. patens* exhibited decreases of 12 and 17% in Middle Meadow and the Gut. Vegetation losses at these sites have been rapid and severe over the last 20+ years (Smith, in press). *S. patens* is not being replaced by *S. alterniflora* in all cases, which results in the formation of un-vegetated, bare areas as *S. patens* retreats landward. *Salicornia* spp. declined by 25% at the Nauset Island site, while *Suaeda* spp. increased by 22% at Hatches harbor (Table 2). While these are seemingly large changes, they probably do not reflect any long-term directional trend since the abundance of annual forbs is influenced by early season rainfall (which affects germination) (Teal and Howes 1996) and the dynamics of seed dispersal (Smith 2007). As such, they tend to be highly variable in time and space.

Table 2. Changes in the frequency of species occurrences by marsh between 2003 and 2008. The dominant perennial species that constitute the vast majority of the vegetation are shaded.

	Distichlis spicata	Juncus gerardii	Limonium carolinianu	Salicornia spp.	Spartina alterniflora	Spartina patens	Suaeda sp.
HH	0%	0%	-4%	4%	-11%	0%	22%
NM	0%	0%	-1%	-3%	-5%	4%	4%
NI	0%	0%	5%	-25%	0%	5%	-10%
MM	0%	-3%	0%	0%	6%	-12%	-6%
WE	0%	0%	2%	5%	0%	2%	12%
Gut	4%	0%	8%	-4%	0%	-17%	-4%
PBay	-3%	-2%	3%	1%	2%	5%	-9%

Vigor of the dominant marsh species – Spartina alterniflora

The mean heights of *S. alterniflora* did not change significantly in any marsh except at the Nauset Mainland site (NM), where values were slightly higher in 2008 (Figure 7) (note: Jeremy marsh had no plots in 2003 and heights were not recorded in the Gut plots in 2003). The reason for this is unclear, but this probably reflects increased vigor of plants in the overwash areas that have since become enriched through stand maturation and the accumulation of organic matter over the last 5 years. In 2003, these plants were growing in almost pure sand. Regardless, in both 2003 and 2008 the tallest plants were found at Hatches Harbor and West End marshes, the shortest at Nauset Island and Nauset mainland. In general, plant heights are related to the flooding regime and drainage characteristics in that *S. alterniflora* is stunted in more saturated, poorly drained areas. In older marshes where the peat is well developed, water retention and sulfide reduction is typically higher than in younger, sandier sediments. As a result, plant heights tend to be much shorter, except along creekbanks which have rapid drainage and, therefore, higher oxygen penetration into the root zone.

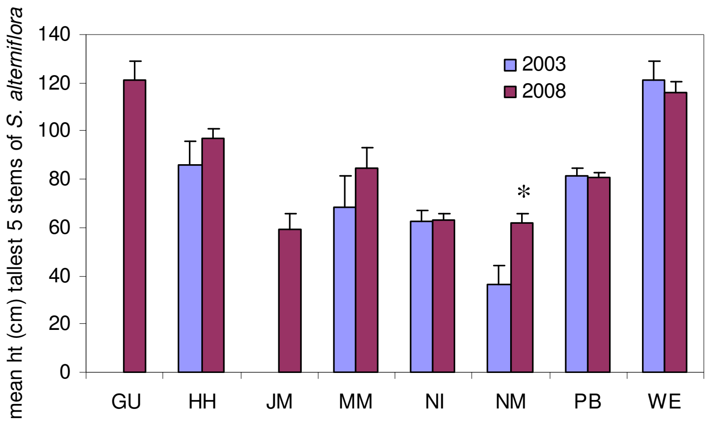


Figure 7. Means of the 5 tallest *S. alterniflora* stems per plot (where it is present) by marsh and year (note: data not collected at the Gut or Jeremy marsh sites in 2003; error bars are standard error of means).

Analysis of aerial imagery (2003 vs. 2007)

With the exception of Middle Meadow, the Gut, and Jeremy marshes, where plant losses (dieback) have continued, comparison of aerial imagery from 2003 (US Forest Service) and 2007 (Digital Globe imagery available on Google Earth) revealed no obvious broad-scale changes in vegetation patterns between 2003 and 2008. In Pleasant Bay, where a break in the barrier beach in 2007 increased the tidal amplitude of the Bay, there is detectable shift landward of the Spring Tide wrack line at the high marsh/upland border. However, major changes in the extent of low vs. high marsh were not conspicuous in the images. In this regard, it should be noted that color infrared (IR) imagery was not available for these dates. With color IR, delineation of the high/low marsh is made easier since the signatures of each community type are more visually distinct than in true color photos. Nonetheless, the imagery proved to be a valuable tool for evaluating how well the ground-monitoring network characterizes the marsh sites and captures temporal change - especially the extent of salt marsh dieback in places such as Jeremy marsh (Figure 8).

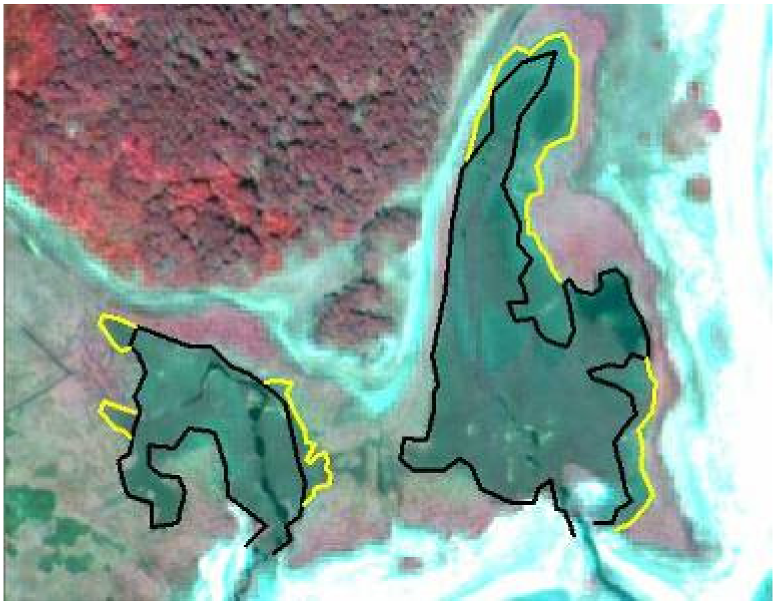


Figure 8. Loss of the high marsh community (primarily *S. patens*; pink signature) in Jeremy marsh between 2003 and 2007 (Quickbird™ imagery). The black line demarcates the position of the seaward edge in 2003, the yellow lines show where this boundary has shifted upslope (landward).

Comparison of all marshes based on 2008 plant community composition

The centroid (mean) values of MDS scores showed that in 2008 the plant communities Pleasant Bay, West End, and Nauset Island marshes were quite similar to each other (Figure 9). The Gut, Nauset mainland, Middle Meadow and Hatches harbor formed an outer ring around this central cluster and Jeremy marsh was quite distinct from all other sites. The relative proportions of the dominant halophytes, *S. alterniflora* vs. *S. patens*, as well as the abundance of *Salicornia* spp., contributed most to dissimilarities among sites.

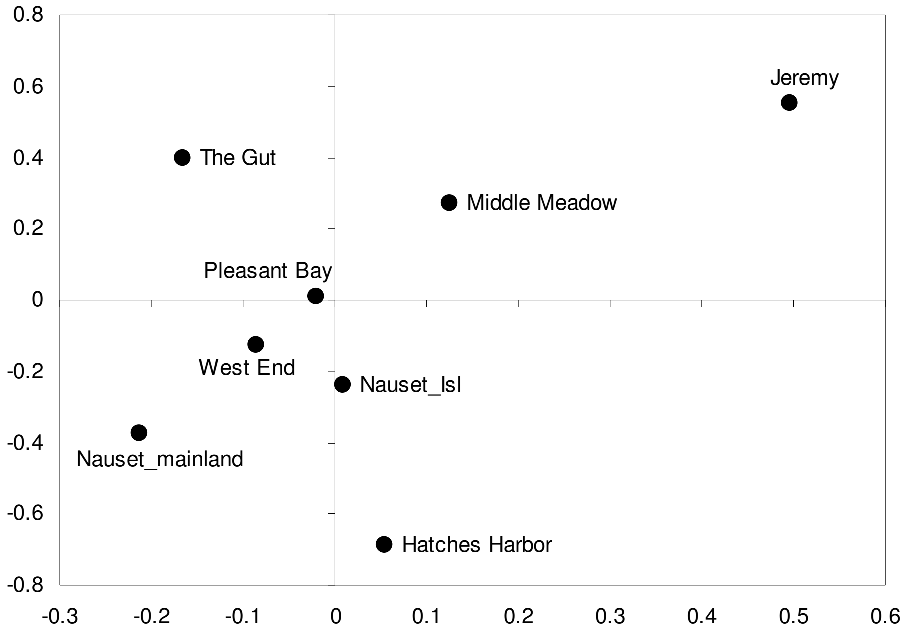


Figure 9. Centroids of MDS values generated from species cover scores.

Halophyte species richness (discounting upland border species such as beachgrass,a seaside goldenrod, etc.) was generally similar among marshes (Table 3). Pleasant Bay stood out in having almost twice the number of species (n=12) as the rest, while Nauset island/mainland had the fewest (n=5). However, the extent to which species richness in the monitoring plots reflects actual species diversity throughout the entire system can be quite variable. Anecdotally, this author has observed species such as *Aster tenuifolius*, *Distichlis spicata*, and *Spergularia salina* while walking around in many of the marshes where they were not recorded in the permanent plots. Pleasant Bay had many more plots, a larger range of porewater salinities, and covered a much broader geographic area than the other marsh sites. These may be the reason for the higher diversity observed there.

Table 3. Species richness (true halophytes only) by marsh (x=present in plot network).

	Gut	HH	JM	MM	NS	PB	WE
Aster tenuifolius						x	
Atriplex hastata	x					x	
Distichlis spicata	x	x	x	x		x	x
Iva frutescens						x	
Juncus gerardii			x	x		x	
Limonium carolinianum	x	x	x	x	x	x	x
Phragmites australis						x	
Salicornia spp	x	x	x	x	x	x	x
Spartina alterniflora	x	x	x	x	x	x	x
Spartina patens	x	x	x	x	x	x	x
Spergularia salina				x		x	x
Suaeda sp.	x	x	x	x	x	x	x
no. spp.	7	6	7	8	5	12	7

Porewater chemistry

Mean salinities exhibited a relatively small range of 30-34 ppt among marshes (Table 4). Salinities within individual marshes varied more, with Pleasant Bay having the largest range among plots (15 ppt) and Nauset Island having the lowest (5 ppt). To some extent salinity ranges reflect elevation ranges. With the exception of hypersaline pannes, which are rare in CACO, salinities generally decrease in an upslope direction.

Table 4. Mean, minimum, and maximum porewater salinities (August 08) by marsh.

	mean	min	max	range	stdev	count	SE
GU	30	24	35	11	3.2	27	0.6
HH	31	25	35	10	2.2	21	0.5
JM	31	29	35	6	2.1	15	0.5
MM	32	25	40	15	3.6	27	0.7
NI	33	30	35	5	1.7	19	0.4
NM	34	30	40	10	2.6	17	0.6
PB	30	20	37	17	5.3	16	1.3
WE	32	25	39	14	3.4	19	0.8

By contrast, sulfide concentrations exhibited extreme variability among and within marshes (Table 5). Sulfide concentrations were highest in Pleasant Bay, which reflects the water-saturated condition of this marsh. In general, the older marshes (Nauset, Pleasant Bay) with thicker peat depth are more prone to sulfide accumulation. Generally speaking, the younger, sandier marshes (West End, Hatches, Gut, Middle Meadow, Jeremy) with lower amounts of organic matter had much lower sulfide levels. In these younger marshes, *S. alterniflora* is typically taller (i.e., more vigorous) due to the absence of toxic sulfide levels.

Table 5. Mean, minimum, and maximum sulfide (µM) concentrations (August 08) by marsh.

	mean	min	max	stdev	count	SE
GU	374	0	3,111	759	45	113
HH	8	0	78	14	38	2
JM	29	0	237	54	23	11
MM	131	0	2,335	410	34	70
NI	1279	0	6,699	1,992	21	435
NM	326	0	2,112	556	18	131
PB	2294	0	14,523	3,084	86	333
WE	210	0	4,146	750	56	100

Comparison of salinities and sulfide concentrations in marshes between 2003 vs. 2008 showed only a couple of significant differences (Figure 10). In both NI and NM sites, salinities were 3-5 ppt lower in 2008. At Nauset mainland this is likely due to increasing vegetation cover. In 2003, when many plots were essentially bare from a recent overwash event, salinities were high since there was no shading effect and evaporation could concentrate salts in the soil. By 2008, when these areas had been grown over, plant shading greatly diminished this process.

Although there were some differences in mean salinity values between years, the overall pattern among marshes was very similar. NM had the highest salinities and Pleasant Bay had the highest sulfides in both 2003 and 2008. Similarly, the Gut and Pleasant Bay had the lowest salinities and the Gut and Hatches Harbor the lowest sulfides in both years.

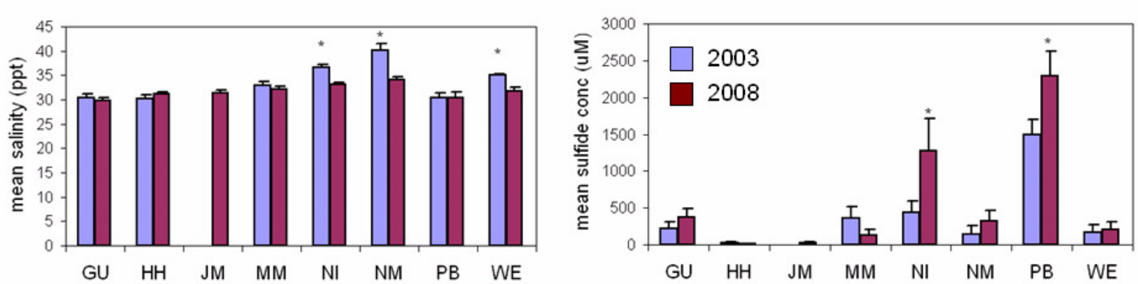


Figure 10. Mean porewater salinities (left) and sulfide (right) concentrations in August by marsh and year (note: porewater samples were not taken in Jeremy marsh in 2003 because this site was not part of the monitoring network at that time; asterisks indicate significant differences among years; error bars are standard error of means).

Special comments on methodological issues and data interpretation

Many years of fieldwork by CACO staff in both tidally-restricted and unrestricted salt marshes have yielded valuable information on the effectiveness of the monitoring program that was begun in Hatches Harbor in 1997. Below are some analyses of the data and methodology and discussion about approaches to monitoring, interpreting information, and suggestions for improvements in the protocol.

Artificial sources of error that can contribute to apparent shifts in species abundance and cover

i) Plot placement - Some of the temporal variability in species frequencies and cover may be artifacts of various procedural problems. Not being able to locate original plot markers is an important issue where heterogeneity in species cover is large. The markers, which are 1.25-inch diameter PVC poles, can be difficult to find because they are hammered in very low to the ground – a necessity to keep them from being bent, broken, and/or carried away by ice. Nonetheless, some markers have been pushed far into the ground by the weight of ice or lost for unknown reasons. Regardless, even very small variations in quadrat placement from one survey to the next can introduce substantial error in places where species coverage is patchy (see Figure 11 below). This in itself is a good reason for monitoring protocols to use permanent locations for sampling plots rather than a fully-randomized design. The coordinates of each plot were originally taken using a Garmin GPS unit (3-4 m accuracy). In the future, these plots should be re-surveyed using a higher quality receiver (Trimble) with sub-meter accuracy.

Figure 11. Vegetation at plot NI3-200 (at marker) in 2003 vs. best guess plot location (field crew couldn’t find marker) in 2008. Note the abundance of *S. patens* in the 2008 photo, which has been determined to be the result of different plot placement given that a 2006 photo of this plot (when the marker was found) shows no *S. patens* whatsoever.

ii) Point-count conversions - A certain amount of error can also be introduced by comparing scores converted from point-count data to those from visual assessments (show example). The former sometimes grossly overestimates or underestimates cover relative to the latter (see diagrams in Appendix I) and these differences are manifested as apparent temporal changes in cover. Reducing the data to species presence/absence and calculating frequencies can provide another way to examine the data without this type of error.

iii) Physical disturbance - Various types of physical disturbance can result in significant changes that has nothing to do with the physiological health of plants. Plots that are close to the edge of tidal channels are often eroded away, especially where creekbank *S. alterniflora* has been lost (Smith et al., in press), and plots near the upper boundary of marshes are episodically smothered by wrack. Ice rafting, barrier-beach overwash events, and sand deposition from adjacent dunes are other factors that may cause major vegetation changes that do not reflect the "health" of the marsh.

Assessment of plot data from the perspective of whole systems

Data from plot-monitoring networks may not reflect important landscape changes occurring within a marsh. In essence, the networks simply do not have the spatial resolution (i.e., enough plots) to detect all landscape-level trends. At CACO there are many examples of this. In Figure 12 below, a large area of *S. patens* disappeared in Middle Meadow, between 2003 and 2007, but the process largely occurred in the area between two monitoring transects, and is therefore not reflected in the plot data.

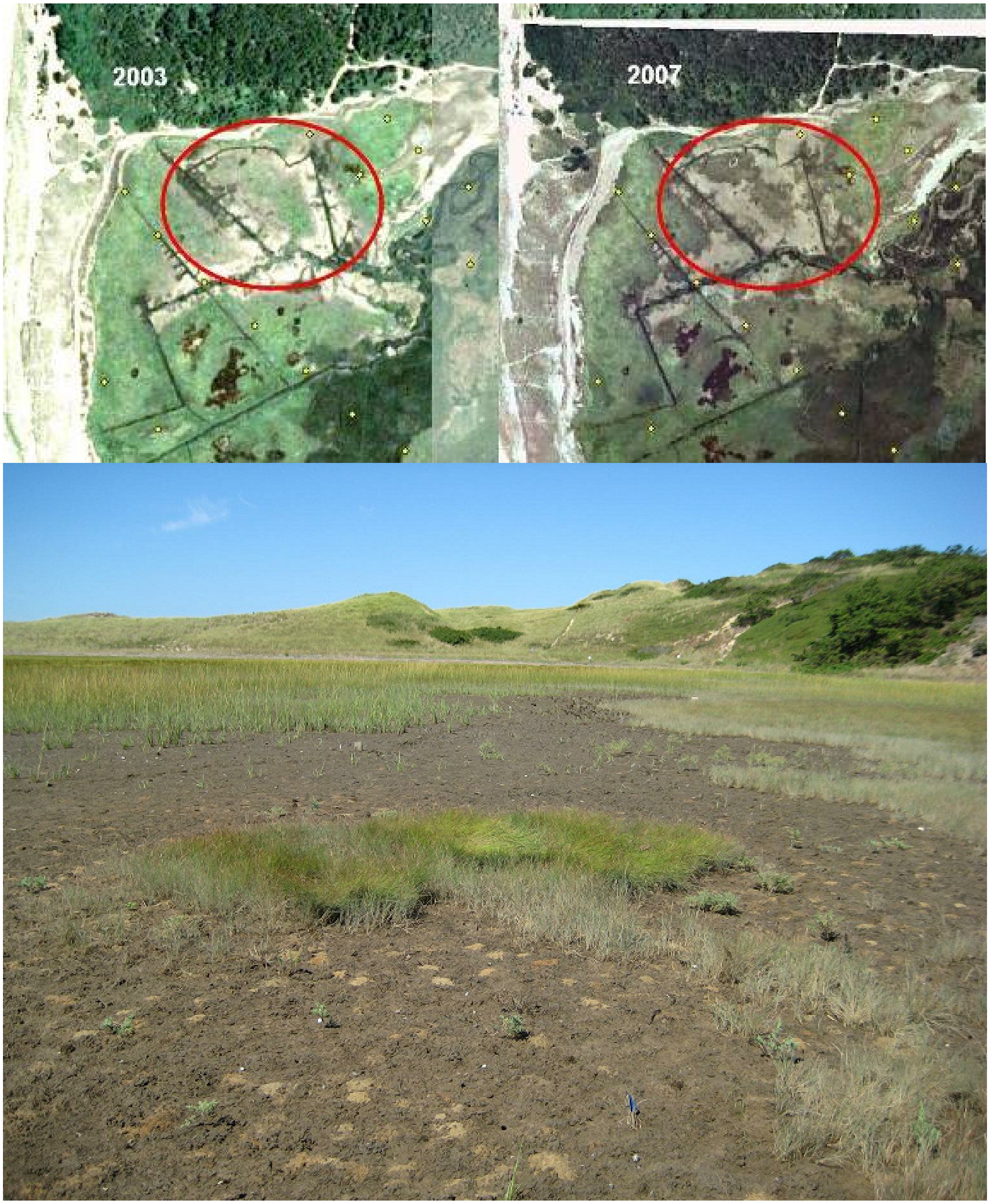


Figure 12. Loss of *S. patens* in an area (circled) not covered by permanent monitoring plots. The photo below shows a ground-level view of the marsh.

Plot networks may also fail to adequately describe changes in low abundance species. *Juncus gerardii* (black rush) provides a good example of this at CACO. This species resides in the upper part of the high marsh zone and is, compared to other salt marsh taxa, intolerant of high salinities and prolonged flooding. As such, it may make for a good indicator of changing tidal regimes – in particular, rising sea level. With sea level rise, this species is predicted to shift landward or disappear altogether. However, as shown in Figure 13, this would never be detected by the existing plot network alone. Fortunately it tends to grow in monospecific patches or bands and is easily distinguishable from other taxa in aerial photos where it has a distinct dark signature. Thus, aerial image analysis provides a good way to track changes in these kinds of communities.

Figure 13. Photo showing locations of *Juncus gerardii* stands (dark signatures within red circled areas) in relation to monitoring plots (yellow circles). Given that only one plot lies within a juncos stand, separate focused monitoring on specific populations may helpful for analyzing salt marsh vegetation trends.

Another "vital sign" for salt marshes is the position of the low/high marsh boundary, which is typically abrupt in New England marshes. This is a critical parameter to monitor since this boundary will shift landward with sea level rise (Donnelly and Bertness 2001). Moreover, it is easier to delineate than the high marsh/upland boundary as the latter is often obscured by steep elevation gradients, wrack disturbances, *Phragmites* invasion, foot paths, and human development which limits landward encroachment. In addition, the low/high marsh boundary is not subject to wave erosion as is the seaward edge of the low marsh. Movement of the former therefore reflects physiological responses to changing abiotic conditions rather than physical forcing. However, monitoring networks may miss key areas where this boundary is shifting rapidly (Figure 14), not to mention that the linear extent of boundary change is impossible to assess with plots along transects.

Figure 14. A nearly 3 ha. area of the Nauset marsh system where considerable area of high marsh (*S. patens* and *D. spicata*; pink signature) has been replaced by *S. alterniflora* (dark red-gray signature). This change has occurred outside the plot network. Inside the plot network *S. patens* in rare.

Plots and changes in marsh geomorphology

Vegetation losses from crab herbivory and other disturbances have resulted in significant changes to tidal creek architecture and marsh area. Once the vegetation is lost, particularly along creekbanks, erosion of marsh sediments is rapid. As a result many tidal creeks have widened and lengthened over the past few decades while the overall area of vegetated marsh has shrunk (Smith, in press) (Figure 15). The total extent of these kinds of losses would be extremely difficult, if not impossible, to estimate from ground-plots – not to mention that plots close to marsh edges would eventually be eroded away during this process.

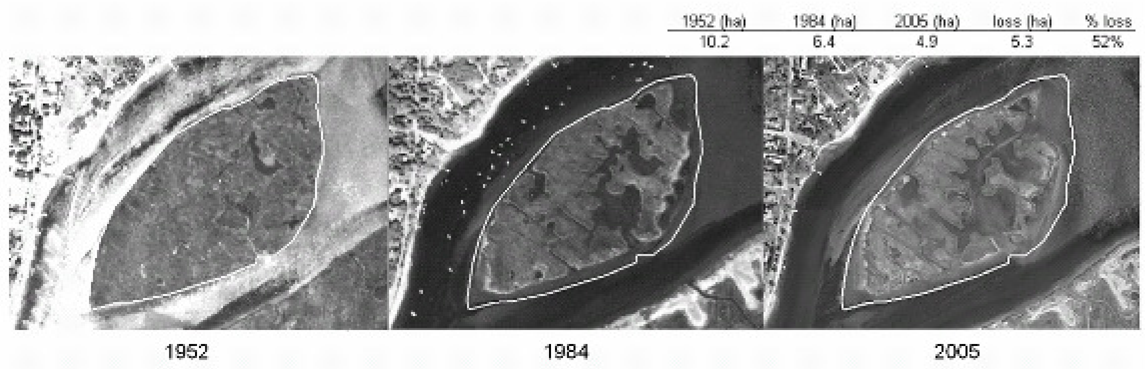


Figure 15. The above photos show the loss of a marsh island between 1952 and 2005 (figure from Smith et al., in press). While a certain number of ground-level plots may show decreases in plant cover, some would be lost entirely. In addition, the spatial intricacies of this kind of change could not be adequately defined along transects

Even when the data from plot networks do manage to capture large temporal changes they may not be representative of real phenomena since even significant changes within numerous plots may be a common occurrence in healthy, normally functioning salt marshes. It is really a question of scale in that changes in 1-m^2 plots may misrepresent what is occurring across the landscape. This is because the boundaries of various species or community types (which are often abrupt in salt marshes) can move and shift while not changing their overall position in the marsh or area extent (Figure 16). Accordingly, very long periods of record would be needed to make sense out of such changes – i.e., as to whether they are proceeding in a certain direction or are simply due to random variability.

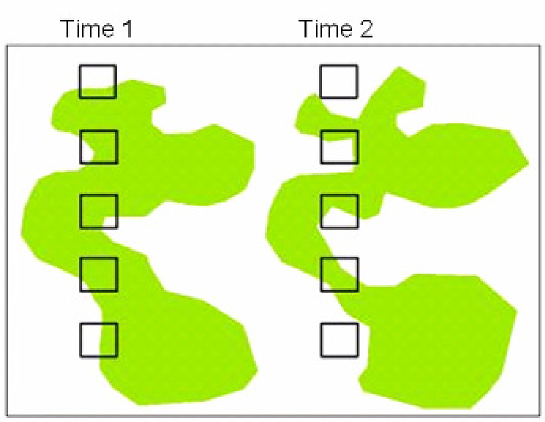


Figure 16. The two green polygons (representing a stand of species X) cover exactly the same area. Because of subtle shifts in shape, a large reduction in the amount of species X within the plot network is detected, although such changes may be ecologically meaningless.

Moreover, small changes in just a few plots can be manifested as statistically significant where the total number of monitoring plots is lower. For example, the frequency *S. alterniflora* decreased by 11% at Hatches Harbor. However, this change is largely the result of losing only a few plants that were present in two different plots. Table 6 is a fictitious dataset that shows the degree to which minor losses can influence species frequency or cover values. Note that the sample size in this example is equal to the minimum number of plots (n=20) recommended by Roman et al. (2001) for monitoring salt marsh vegetation in unrestricted marshes. The statistical remedy is to have a very large number of plots in each marsh area, which necessitates the use of visual methods over point counts. However, the expansion of monitoring networks quickly reaches a limit due to the many logistical issues associated with increased sampling.

Table 6. Fictitious datasets showing how 1) frequencies or 2) mean % cover values in *Spartina alterniflora* (SA) (calculated from mid-points of cover class categories) can differ substantially based on changes in just 2 out of 20 plots.

	presence/absence		cover class		mid pt %	
	SA (yr1)	SA (yr2)	SA (yr1)	SA (yr2)	SA (yr1)	SA (yr2)
1	0	0	0	0	0	0
2	0	0	0	0	0	0
3	1	1	5	5	37.5	37.5
4	1	1	7	5	87.5	37.5
5	1	0	2	2	3	3
6	1	1	1	1	0.5	0.5
7	1	1	1	1	0.5	0.5
8	0	0	0	0	0	0
9	1	0	5	3	37.5	8
10	0	0	0	0	0	0
11	0	0	0	0	0	0
12	0	0	0	0	0	0
13	0	0	0	0	0	0
14	1	1	3	3	8	8
15	0	0	0	0	0	0
16	0	0	0	0	0	0
17	0	0	0	0	0	0
18	0	0	0	0	0	0
19	0	0	0	0	0	0
20	1	1	4	4	17.5	17.5
freq	40%	30%		means	9.6	5.6

Placement of new plots – random vs. uniform distances along transects

In 2008, new plots were added to CACO's monitoring network in a different manner than had been done previously. Based on statistical recommendations from the Northeast Coastal and Barrier Network (NCBN), plot locations were randomized along randomly-placed transects instead of uniformly-spacing them along randomly-placed transects. While the statistical theory behind this approach is recognized, it can result in a sampling layout that seriously compromises the ability to detect change along gradients. In salt marshes, elevation gradient is an important factor regulating species distributions. Since elevations generally increase toward the upland boundary, uniformly-spaced plots do a decent job capturing seaward or landward shifts in species distributions. While randomly-spaced plots can achieve the same result, it is well within the realm of possibility that randomization generates a layout with very large distances between certain plots. In such circumstances, even large changes (many tens of meters) in the position of an ecotone may go undetected for very long periods of time (Figure 17).

Systematic plot placements (along transects that are randomly located) has been the approach taken by Roman et al. (2001) and many others as a way to ensure that such changes are detected. Thus, the systematic approach be more useful in long-term monitoring programs despite ongoing statistical arguments about treating plots as independent replicates in this design.

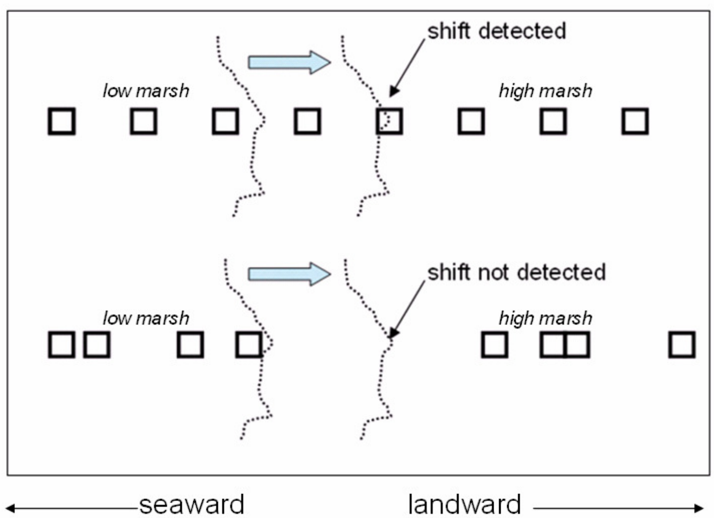


Figure 17. Uniform (top) vs. randomized (bottom) plot placement along transects. Note the possibility of the latter design to fail to capture changes in ecotone shifts (e.g., landward shift in the seaward edge of the high marsh).

Because of the abovementioned shortcomings in ground-level monitoring programs and many ways to misinterpret the data, aerial image analysis is critical for overall perspective. By analyzing aerial photographs, one can assess whether the data reflect important, broad-scale processes or are highly localized or spurious phenomena. As such, the most important function of ground-level plots may be to provide critical information for ground-truthing aerial photo signatures.

Porewater sampling – sample size vs. frequency

The salinity and sulfide "snapshots" discussed previously are useful for general characterizations of marshes. However, they have questionable value for long term monitoring given the level of temporal variability that can occur within a single growing season. Precipitation, evaporation, and tidal cycle all exert considerable influence on these variables. Accordingly, it may be more information to select a much smaller number of key sampling locations in marshes where porewater can be monitored at a higher frequency. For example, one could set up permanent sampling locations at the seaward edge of the low marsh, the high-low marsh boundary, and upland edge. Subsequently, monitoring could be conducted throughout the extent of the growing season in order to better quantify salinity and sulfide regimes. In the future, if affordable salinity loggers become available, this would be the preferred option for long-term monitoring of this variable.

Role of experimental research

One final point about interpreting monitoring data is that focused, experimental research will usually be needed to properly analyze changes in salt marsh ecosystems and especially to predict future conditions. Monitoring data is helpful in informing us about what is happening, but it frequently fails to explain why it is happening given the multitude of confounding factors in the natural environment. Only when specific variables are controlled while others are intentionally manipulated (i.e., experimental hypothesis-testing) are we able to gain insight on mechanisms of change.

An excellent example is the abovementioned research lead by Dr. Mark Bertness on *S. alterniflora* losses through crab herbivory (Holdredge et al. in press). Understanding the true nature of this phenomenon has been instrumental in explaining past changes in marsh structure and function, in projecting the future trajectory of these systems, and in developing management action plans (crab depletion experiments will be conducted in 2009). Ongoing research by NPS staff on the causes of high marsh dieback (*S. patens*) is equally as important. In fact, research is in many ways more proactive than monitoring. Although both are integral components of assessing the health of ecosystems, researching the questions generated by monitoring may ultimately provide the best answers on how CACO should manage its resources.

Recommendations for improving CACO's salt marsh vegetation monitoring protocol

1. Aerial image analysis should be incorporated into the monitoring protocol so that key characteristics of marshes (e.g., movement of the low/high marsh border) can be tracked.
2. Visual cover class estimates should be used instead of point counts so that many more plots can be sampled and, therefore, spatial coverage improved.
3. Plots should be GPSed using an instrument (e.g., Trimble™ GPS) with sub-meter accuracy in the event that physical markers cannot be found.
4. It may be helpful to incorporate intensive monitoring of certain key species (e.g., *Juncus gerardii* populations) for added value.
5. For porewater sampling, it may be better to focus on key sites within marshes (i.e., low/high marsh transition zone) for more intensive (higher frequency) monitoring.
6. Experimental research should be incorporated as needed for interpreting monitoring data.

Acknowledgements

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Appendix I. Justification for using visual estimates of cover class rather than the point-intercept (also known as point-count) method.

1. In tall, dense woody vegetation (including *Phragmites*) it is extremely difficult to set up the plot and grid (dividing rods) system, not to mention threading the bayonet through the vegetation in order to touch each point on the rod. This is particularly a true when attempting to collect data from the middle of a plot, which requires leaning over it – something that cannot physically be done in some plots without moving a lot of vegetation out of the way. In fact, the simple act of positioning the bayonet in the right place often ends up upsetting the position of the rods on the ground (which are really some distance up off the ground, laying on the bottom parts of stems, etc.), not to mention breaking delicate woody stems. When working in vegetation with a lot of thorny or noxious plants (e.g., rose and blackberry shrubs, poison ivy), setting up the plot can be hazardous at best, miserable at worst, and extremely time consuming in all cases.

2. When the vegetation is tall (e.g., overhead) you have to use an extremely long bayonet to figure out whether overstory vegetation is touching it or not. Regardless, you are just "eyeballing" hits or misses. Also, it is problematic to hold the bayonet in a true vertical position, which ultimately affects whether you call a "hit" or not.

3. The point-intercept method calls for using a bayonet that is ≤ 3mm in diameter but we have not been able to find a suitable material of this size that will not end up getting bent during fieldwork. Moreover, the longer the bayonet required the greater chance that this will happen.

4. Bayonet width is also a critical factor - the thinner the bayonet, the more likely you are to miss touching the vegetation.

5. When the wind is blowing it is very difficult, if not impossible, to tell whether vegetation would normally be touching the bayonet or not. In other words, this is another point at which you're making a subjective guess.

6. When the bayonet is extremely close to a plant (i.e., < 1-2 mm) it is very tempting to call a hit, especially if there is significant cover of that species, but no hits have been called yet.

7. It is possible to seriously overestimate the cover of a species. For example, a thin bladed leaf (such as *Ammophila brevilugulata*) can be recorded multiple times, resulting in 2 or 3 hits, which translates to 4-6% cover. However, the actual area of that leaf relative to the $1m^2$ plot can be substantially lower than 1%.

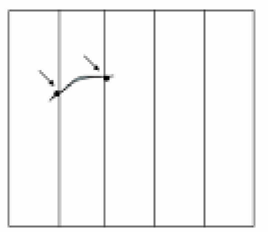


Figure above shows how a single leaf can account for 2 hits (=4% cover), even though true area cover is < 1% of the total area.

8. It is also possible to seriously underestimate the cover of plants, particularly species that have dense clustering of stems and don't have spreading leaves, but still comprise a large amount of cover (see Figure below for an example).

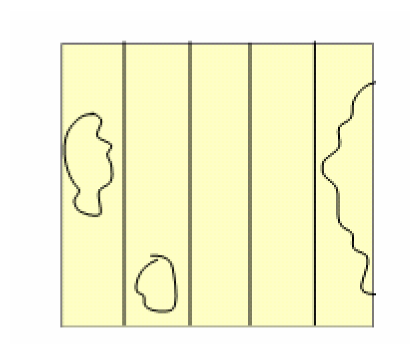


Figure above shows how vegetation (polygons) between rods can comprise a large amount of cover but go "hitless" in the point intercept method.

9. The percent cover values from point intercept are converted back to cover class values for statistical analysis by ANOSIM anyway.

10. Point intercept requires a huge quantity of data to be recorded for each plot. At minimum, the field assistant must record 50 bits of information. For multispecies plots, this number can rise to well over 100. In this, there is ample opportunity for error during both the recording and transcription process.

11. Vegetation in salt marshes can change substantially over the course of 2-3 weeks, especially annual species which can germinate throughout the growing season and grow quickly. Therefore, the large amount of time it takes to conduct point intercept data incorporates more temporal variability because the sampling window is so large. For example, it takes approximately 3 weeks to sample Hatches Harbor by point intercept and about 2 months to cover the entire salt marsh plot network at CACO. With the visual

cover method, all plots within the entire network can be assessed in a very short timeframe (approximately 6 or 7 days).

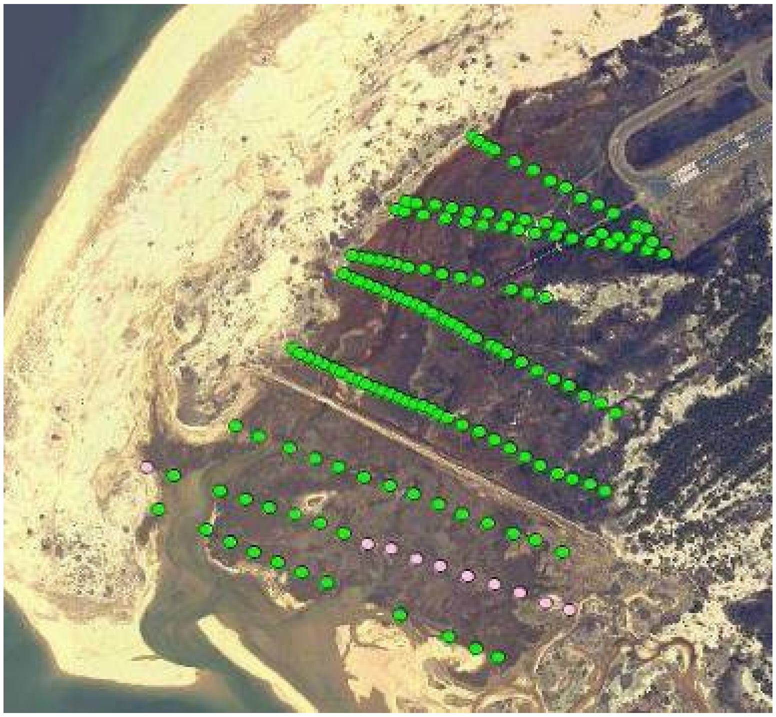
12. Because the time it takes to assess vegetation by cover class estimates is so short relative to the point intercept method, the data can be collected closer to the end of the growing season (end of August/September) when plant species composition and biomass have stabilized for the season.

13. Visual estimation of cover class, while inherently having some level of subjectivity, is a well-established and accepted method of monitoring vegetation. With good instruction and sufficient time for practice, observers can develop the skills necessary to make accurate estimates. Regardless, some level of observer variability in cover class estimates is not necessarily a bad thing because it follows that any statistically significant trends that emerge from the data are likely to reflect important and potentially broader-scale processes. In other words, data that show significant directional changes in cover have to "overcome" a certain level of observer variability.

Overall conclusion:

The point intercept method is highly subjective in many ways as well as logistically cumbersome and time consuming. The point intercept method requires two people whereas visual estimates of cover can easily be done by one person in approximately 1/10 the amount of time. By using visual cover estimates, a larger quantity of data can be collected in a much shorter period of time and more frequently using visual cover class estimates.

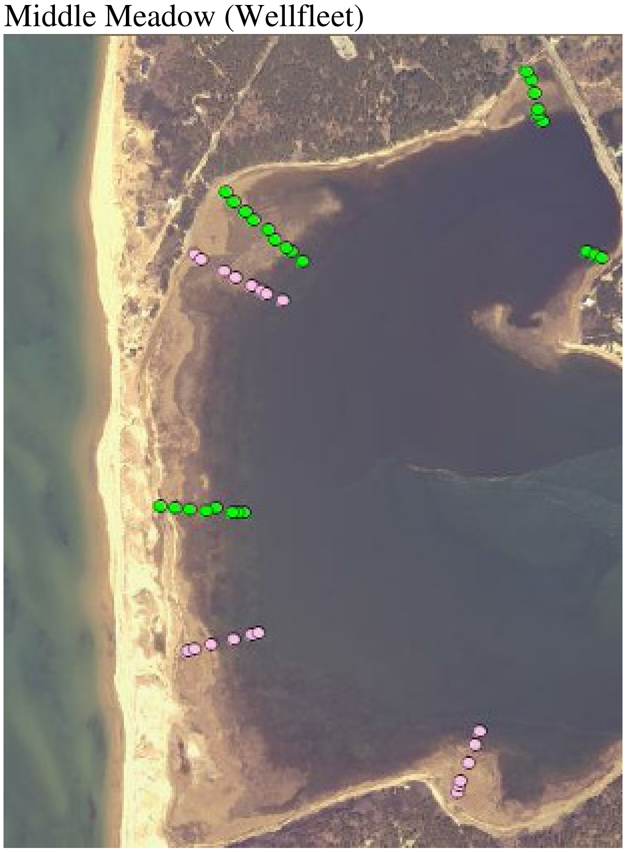
Appendix II. Maps of marsh sites with original (green) and additional (pink) plot locations**.**



Hatches Harbor (Provincetown)

West End (Provincetown)

Middle Meadow (Wellfleet)



The Gut (Wellfleet)

Jeremy marsh (Wellfleet)

Nauset marsh “island” (bottom transects) and “mainland” (top transects) (Eastham)

Pleasant Bay (Orleans/Chatham)